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## **A framework for valuing spatially targeted peatland restoration**

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### **Abstract**

Recent evidence suggests that the degree of degradation of peatlands is substantial, and that there is a significant potential to enhance the delivery of a wide range of ecosystem services by investing in peatland restoration. However, little is known about the social welfare impacts of peatland restoration and in particular how to spatially target restoration activities to maximise net benefits from investments in restoration. This paper investigates the steps required to conduct a spatially explicit economic impact assessment of peatland restoration, and highlights and discusses key requirements and issues associated with such an assessment. We find that spatially explicit modelling of the biophysical impacts of restoration over time is challenging due to non-linear effects and interaction effects. This has repercussions for the spatially explicit assessment of costs and benefits, which in itself is a demanding task. We conclude that the gains of investing in the research needed to conduct such an assessment can be high, both in terms of advancing science and in terms of providing useful information for decision makers.

**Keywords:** Peatland restoration; Ecosystem services; Spatial targeting; Valuation; Cost-Benefit analysis

## 1 Introduction

Peatland restoration can, under certain conditions, act as a greenhouse gas (GHG) sink, thus generating benefits in terms of GHG emission reductions (Peacock et al., 2013; Quin et al., 2014). Restoration has also been found to enhance the delivery of other ecosystem services (ES) such as erosion control (Wilson et al., 2011), and ES related to water quality (Joosten et al., 2012), recreation and biodiversity (D'Astous et al., 2013)<sup>1</sup>. Recently, peatlands have received much policy attention for their contribution to climate change mitigation and the potential of peatland restoration to achieve national emission reduction targets cost-effectively (Bain et al., 2011; ASC, 2013). Peatland restoration has indeed been globally recognised for its potential role in contributing to international climate change mitigation (Kyoto Protocol) and biodiversity conservation (Ramsar Convention on Wetlands; Nagoya Protocol of the UN Convention on Biological Diversity) commitments (Bonn et al., this issue). It has also been suggested that peatland restoration can potentially contribute to compliance with the EC Water Framework Directive (Martin-Ortega et al., this issue).

Given this policy interest in peatland restoration and conservation, limited restoration budgets and the spatially-varying costs of restoration, there is a need for information on how to spatially prioritise restoration and conservation efforts to deliver the greatest welfare gains to society. That is, there is a need to identify what should be done and where<sup>2</sup>. This paper contributes to a better understanding of the economics of peatland ES and identifies some of the key challenges that need to be addressed for prioritising peatland restoration and conservation<sup>3</sup> activities. In this article, we focus on peatland ecosystems in the UK, which contain over half of the UK's soil carbon store (Defra, 2009). However, the conceptual points and empirical methods referred to in this paper are also applicable to other geographic regions and ecosystems where peatlands are found globally, such as South-East Asia.

From an economic point of view, the main question is whether restoring peatlands increases overall social welfare, and if so, where and how the costs and benefits are generated. In this paper, we propose that Cost-Benefit Analysis (CBA), which requires all costs and benefits associated with restoration to be identified and measured in monetary terms, would be a useful approach to evaluate

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<sup>1</sup> Here, ES are defined as those outputs of the ecosystem that contribute directly to human well-being (Fisher et al., 2009).

<sup>2</sup> For example, the 2013 Green Stimulus Peatland Restoration Project in Scotland (<http://www.snh.gov.uk/climate-change/what-snh-is-doing/green-stimulus-peatland-restoration/>), with a budget of £1.7 million over 2 ½ years, illustrates the direct policy relevance of these questions.

<sup>3</sup> Reflecting the widespread degradation of UK peatlands, in the remainder of this paper, we – unless made explicit – refer only to restoration, i.e., improving the condition of previously degraded peatland, and related activities. The aspects covered in this paper equally apply to conservation, i.e., preserving peatland in good condition, and related activities. The main difference is that conserving sites in good condition may incur lower initial capital costs. Note that, within the UK, protected sites may be degraded (Bain et al., 2011). In fact, a considerable amount of effort has been dedicated in the past to restore degraded and protected Sites of Special Scientific Interest (SSSI) (Worrall et al., 2011).

peatland restoration. In particular, CBA offers a framework within which discrepancies between private and public costs and benefits can be accounted for explicitly.

Identifying those peatland areas with the greatest welfare gain from restoration requires recognition of a wide range of ES associated with this habitat. Besides changes in GHG emissions, peatland restoration may have beneficial impacts on a range of biodiversity indicators, food and timber provisioning services, recreation (e.g. via access and aesthetics), wild-fire risk, and water-related services (e.g. flood risk and water quality) (Bain et al., 2011; Tinch et al., 2010; Evans et al., this issue; Martin-Ortega et al., this issue).

The key contribution of this paper is to identify and discuss the multiple challenges for conducting an economic analysis that would allow peatland areas to be prioritised or targeted for restoration with regard to the social net benefit criterion (Hanley and Barbier, 2009). These challenges are related to the assessment of costs and benefits of peatland restoration over *space* and *time*, and the distribution of these costs and benefits across stakeholders at different spatial scales. CBA facilitates identification of and allowance for price distortions arising from, for example, subsidy transfers to certain land uses and the non-market values of environmental impacts. This allows discrepancies between private and public costs and benefits to be accounted for, and thus estimation of net values to society.

An important aspect of an ES approach should be the consideration of spatial variation in ES provision and benefits (see Bateman et al., 2003, 2009; Turner et al., 2010). Specifically, the questions arising from spatial targeting of peatland restoration require the consideration of spatial variability in ecological response relationships and resulting benefits arising from changes in ecosystem quality; dependency of economic values on the spatial distribution of beneficiaries; and spatial variation in (opportunity) costs of restoration. Furthermore, the paper discusses how any analysis aimed towards the spatial prioritisation of restoration activities may be conditioned by aspects of risk and uncertainty.

## **2 State of UK peatlands, current management and restoration**

Evidence on the current state of UK's peatlands is limited and has been summarised in Bain et al. (2011). According to the best available evidence (JNCC, 2011; Littlewood et al., 2010), less than 20% of the UK's peatlands are largely or entirely undamaged. Many peatlands are eroded, modified or destroyed through extraction or conversion to other land uses. Recent reviews of the ecological condition of peatlands in the UK (JNCC, 2011) suggest that the extent of disturbance and degradation is considerable for both nationally and internationally protected sites. There is thus much peatland that could potentially be restored. Given the scale and cost of this task, there is therefore an urgent need to provide information that can be used for prioritisation of restoration activities across the UK.

While anthropogenic external pressures such as climate change, nitrogen and sulphur deposition contribute to peatland degradation, management of peatlands in pursuit of a range of land management objectives is the largest source of degradation. Current management of peatlands includes livestock grazing with sheep and cattle, agricultural cultivation, and managed burning (JNCC, 2011). Heathlands are also managed for grouse and deer populations for shooting. Historically, many peatlands have been drained in order to increase returns to agriculture and game management. This drainage produces a range of environmental side effects, including oxidisation and shrinkage of peat, with associated increases in carbon losses (Rawlins and Morris, 2010).

Suggested restoration practices include changes in land management (e.g., cessation of moorland burning, reducing stocking density to prevent overgrazing); blocking of ditches to initiate re-wetting, which requires localised engineering interventions; and more intensive interventions associated with deforestation and re-vegetation of bare peat. Several restoration practices may be combined at a single site.

Peatland sites vary considerably in their characteristics - for example in terms of the size and density of drainage ditches or the proportion of bare peat present. This means that different restoration activities will be required for different sites. Also, the required volume and intensity of a given restoration activity and/or the combination of activities will vary by site (Lunt et al., 2010; Tanneberger and Wichtmann, 2011). For example, a lightly degraded site may require only limited ditch blocking whilst a more heavily degraded site may require more extensive ditch blocking but also stabilisation and re-vegetation of bare peat. Such differences lead to variation of restoration costs across sites. The costs fall into three broad categories - capital costs, on-going management costs and opportunity costs<sup>4</sup>. Opportunity costs reflect the displacement of whatever ES are derived from a peatland site prior to restoration, which in many cases will be commercialised provisioning services (e.g. livestock, timber or grouse production) including negative externalities associated with agricultural production. Such costs may be negative from a private and/or social point of view, if upland farming generates negative profits once subsidies and off-farm income are excluded (Ács et al, 2010). Restoration may or may not be associated with the continued delivery of commercialised provisioning services, depending on the degree to which production activities are displaced. Extensive grazing or grouse moors might be compatible with restoration in some cases, and indeed a sustainable agricultural use of rewetted peatlands could be an important option to consider (Joosten et al., 2012).

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<sup>4</sup> At a practical level, some restoration projects face land acquisition costs – but these may be interpreted as the capitalised value of opportunity costs. That is, rather than paying annual compensation to a land owner for income forgone, ownership of the land is bought outright.

### 3 Economic impact assessment of peatland restoration: general framework

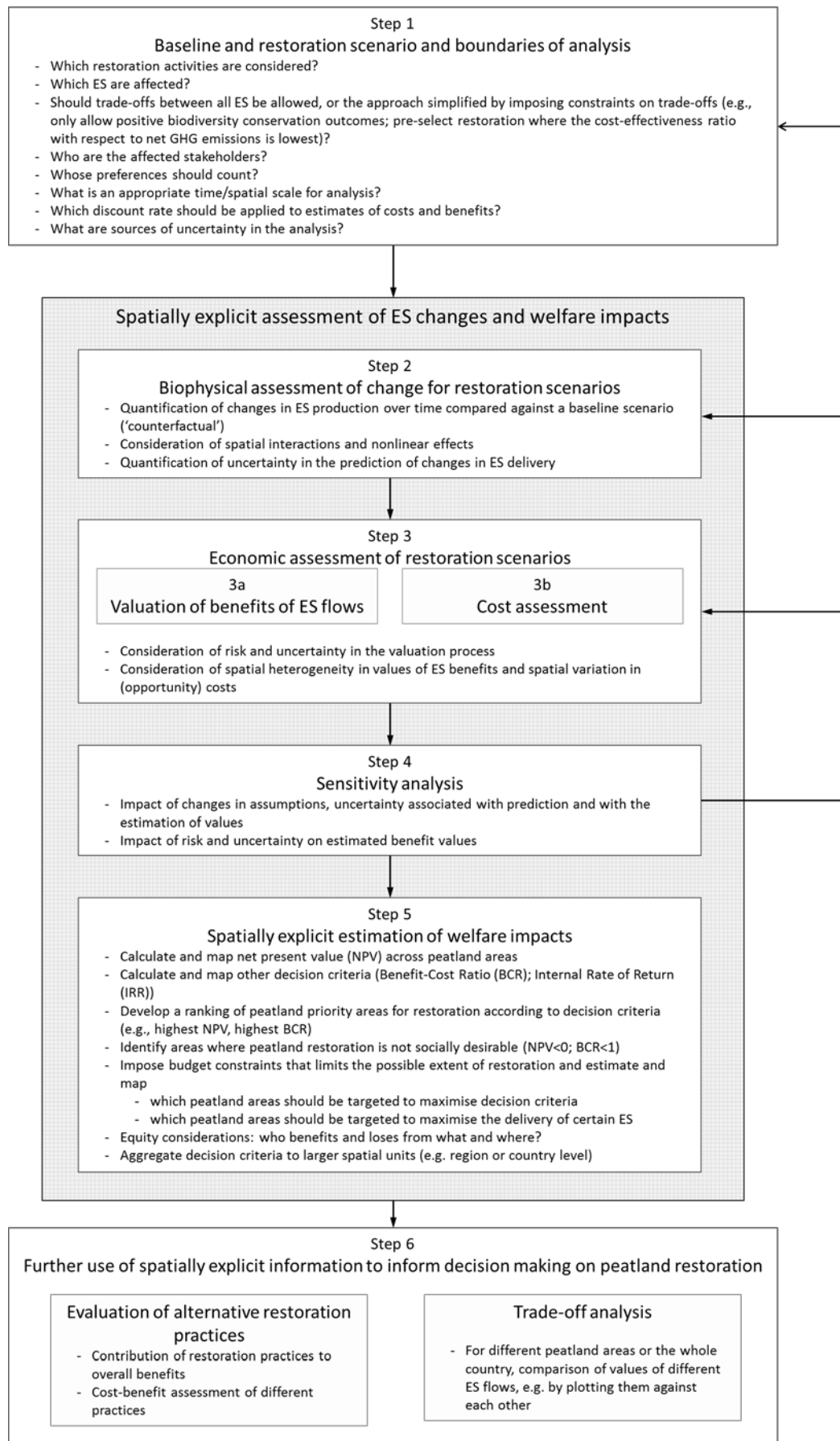
The general Cost–Benefit Analysis (CBA) framework for analysing changes in land management has been widely described and applied; for recent examples see Pattison et al. (2011) and Cortus et al. (2011). CBA enables the comparison of alternative courses of action, including the option to do nothing. It rests on the monetary valuation of all impacts of alternative courses of action, including valuation of the welfare impacts of changes in ES provision (Hanley and Barbier, 2009). It accounts explicitly for market distortions caused by, for example, market imperfections, externalities, taxes and subsidies to facilitate estimation of the net value to society and the trade-offs involved. This paper identifies key challenges for CBA aimed at informing the spatial prioritising or targeting of peatland restoration.

Figure 1 gives an overview of the steps associated with a CBA of peatland restoration activities, which will be discussed in greater detail in this paper. First, the boundaries of analysis need to be defined and scenarios of change need to be developed (Step 1). Scenario development involves projecting changes in peatlands into the future, both in case of business-as-usual (baseline scenario) and in case of additional restoration activities. The changes in peatlands are then assessed with respect to impacts on ES provision levels over space and time (Step 2). That change in provision results from a change in how the ecosystem is managed, but will have to consider changes in external conditions (e.g., related to climate change, agricultural prices or policy drivers). ES provision levels (flows) under a restoration scenario are then compared to a baseline or counterfactual state of the world. The economic assessment (Step 3) includes monetary valuation of both the benefits (Step 3a) and costs (Step 3b) related to ES provision to estimate social welfare *changes* resulting from a *change* in ES flows due to additional restoration relative to a business-as-usual scenario. Cost-benefit estimates will be sensitive to: i) changes in assumptions made, for example when defining the scenario, population of affected parties or physical boundaries for analysis; ii) changes in outcomes of biophysical assessments of the impacts of land management (e.g., errors in biophysical models, changing in assumptions underlying these models); and iii) assumptions and measurement errors associated with the valuation process (see “sensitivity analysis”, Step 4).

Spatial targeting of peatland restoration requires an understanding of the spatial distribution of costs and benefits. Spatial variation in costs and benefits of peatland related ES is caused by spatial variation in the biophysical, social, economic and political characteristics of peatlands and their beneficiaries (the human population). For example, the same restoration technique does not always result in the same outcomes in all locations. The difference in response is predominantly due to the spatial and temporal heterogeneity inherent in peatlands (Parry et al. 2014). Spatial variation in costs and benefits is also affected by the way ES flow across a landscape and people are able to travel and

enjoy peatlands, i.e. the ecosystem/human interactions. Both ecological and economic models should therefore aim to be spatially explicit (Fisher et al., 2011; Balmford et al., 2011; Bateman et al. 2011a).

To derive a ranking of peatland priority areas for restoration that can be used for targeting, spatially explicit information on the flow of costs and benefits over time can be used to estimate Net Present Value (NPV) and Benefit-Cost Ratio (BCR) for different peatland areas. As a decision rule, those peatland areas with the highest NPV and BCR should be given priority for restoration, since their restoration reflects the greatest social welfare gain within a certain time period. A negative NPV or a BCR that is less than one suggests that restoration of a particular peatland area is not socially desirable. There will be constraints on the supply of (public) funds to support restoration activities. In such a case, spatially explicit information on NPV and BCR can be used to estimate and map which peatland areas should be targeted, and which proportion of the total peatland area should be restored. Those solutions that are best from an economic efficiency point of view can also be screened for equity implications, i.e. in terms of who benefits and who loses from the restoration of a peatland area. Additional uses of the spatially explicit analysis include an evaluation of restoration practices with respect to their overall contribution to social welfare; and an assessment of ES trade-offs between different areas of peatland.



**Figure 1** Steps involved in conducting a spatially explicit CBA of peatland restoration



## 4 Assessing changes in peatland condition and associated ES flows: basic considerations

### 4.1 Scoping and scenario development

Peatlands deliver a wide range of ES, and restoration will affect the level of provision for these service flows. The main ES obtained from peatland areas are carbon sequestration, biodiversity<sup>5</sup> and aesthetic values, water supply (or quality) and flow regulation, recreation, shooting, agricultural crops and animal products, and raw material (peat) (Wichmann et al., in press; van der Wal et al., 2011; Bonn et al., 2010). The relative importance of these services is likely to differ across peatlands, and as part of a scoping and boundary setting exercise, the most important services should be identified at the local level of the case study.

For an impact assessment of peatland restoration, future scenarios of changes in peatlands related to restoration need to be compared to a baseline scenario, which involves future projections for the case of no *additional* restoration activities and is referred to as the counterfactual. The baseline scenario is key to assess the cost of (policy) inaction in terms of peatland restoration. Information needs include spatially explicit knowledge on peatland distribution and condition, information on past and current land use and management (for example regarding drainage) and existing restoration activities.

Alternative future scenarios for peatland ecosystems can be generated as narratives, or storylines, or based on ecosystem models that can project future development. Creating scenarios of future change requires the definition of a set of exogenous drivers of ecosystem change. These can include environmental drivers such as atmospheric deposition resulting in acidification, and climate change. Furthermore, it is possible to include changes in social dimensions such as population sizes and profiles. Scenarios may also be defined according to a set of policies in response to changes in exogenous social or biophysical drivers (Hanley et al., 2012). For example, population growth will increase land prices for housing and raise food demand, and both developments may increase pressure on green space. Climate change and food security policies may therefore affect political support for conversion of agriculture to peatland in opposite directions. Literature on the future of areas that are rich in peatlands may also provide useful guidance (for example, Reed et al. 2009).

The development of restoration scenarios requires the definition a set of potential activities or measures that seek restoration, including information on management intensities. For example, a range of generic restoration options (e.g. re-wetting, ditch-blocking, gully blocking, stocking density reduction) may be readily identified yet may be deployed with different intensities or in different

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<sup>5</sup> We recognise that biodiversity has multiple roles in an ecosystem service framework. It contributes to and regulates ecosystem processes and functioning, thereby underpinning the provision of a wide range of services valued by humans. Here, we refer to aspects of biodiversity directly enjoyed or consumed by humans (i.e., biodiversity as a final service *sensu* Fisher et al. (2009) or good in line with the definitions provided in the UK National Ecosystem Assessment). See Mace et al. (2011) for an attempt to characterise the role of biodiversity in an ecosystem service framework.

combinations at sites that differ in current land use and ecological condition. These factors together determine the effectiveness of different restoration measures, and thereby the change in ES provision that can be achieved. The evaluation of a larger number of restoration activities, and differentiation between their intensities adds to realism, but also increases the complexity of an analysis aimed at identifying the optimal choice of activities across peatland areas.

In the assessment of restoration policies, it is desirable for creating consensus and legitimacy to involve stakeholders, a step that requires an identification of people who are affected by changes in peatland management and ecosystem services. Understanding who may gain or lose under various scenarios will provide insights into the social and economic factors that influence current land management values as well as preferences for future developments (Rawlins and Morris, 2010). Cost-benefit Analysis (CBA) typically looks at overall social welfare impacts in order to select the option that generates the highest social welfare, which implicitly assumes that some mechanism could be put in place whereby ‘winners’ can compensate ‘losers’. Taking this a step further, for example in a Kaldor-Hicks tableau format (Krutilla, 2005), the welfare impacts of restoration may be listed for different stakeholder groups to identify who wins and who loses. If disparities between stakeholder groups occur, it is likely that changing management plans will meet opposition at some level, and some compensation mechanism (redistribution of benefits) may be necessary from an equity point of view.

#### 4.2 Time scale

Ecological responses to restoration will vary depending on the current state of the peatland, and the response will often only be noticeable after a few years, where different starting states and ES will follow different time paths of ES delivery. For example, for GHG emissions, the time required to achieve significant emissions reduction will vary from a few years in the case of less severely damaged peatlands, to more than a decade for more heavily damaged peatlands. However, in the case of a heavily damaged peatland, current and projected GHG emissions are higher compared to a slightly degraded peatland, so the potential savings that are eventually realised will be higher (Artz et al., 2012). This clearly illustrates the importance of differentiating between different current peatland conditions, and precludes the use of single ‘one-fit for all’ estimates with respect to the ES response related to different restoration activities.

The chosen time frame of the analysis may impact upon the overall outcome of the CBA, especially when impacts are non-linear over time. The timeframe should ideally be set to allow for the ecosystem and its ecological functions to arrive at some state of saturation or equilibrium condition with respect to ES flows. Otherwise, there is a risk that major (positive or negative) changes in ES flows over time get ignored. For example, rewetting of previously drained peat can initially lead to

increased methane emissions ('methane spike'), thereby temporarily increasing the global warming potential (Couwenberg and Fritz, 2012; Cooper et al., under review). Over time, however, rewetting will result in a net reduction of net GHG emissions compared to business-as-usual (Couwenberg et al., 2011). It is also uncertain whether peatland restoration would eventually result in full restoration of ES comparable with a near-natural reference site (Moreno-Mateos et al., 2012).

Similarly, the distribution of ES flows over time needs to be understood for the non-restoration scenario. Longer time frames may result in predictions of non-marginal (system) changes and the almost complete loss of ecological functioning as a foundation for providing valuable service flows. Importantly, complete degradation of peatlands may essentially be considered irreversible, and long time frames increase the risk of hysteresis (i.e. that restoration of ES provision may only be achieved at a huge cost) (Carpenter et al., 1999). This may be problematic in a valuation context, especially when the spatial scale of analysis is large: ES values derived from environmental valuation techniques normally apply to marginal (and at least theoretically reversible) changes. Another consideration is that the variability in model predictions changes, and often increases, when the time period covered by the analysis increases. Regardless of the approach chosen to predict future conditions and ecosystems service impacts for the baseline scenario and the restoration alternatives, uncertainty with respect to outcomes can be expected to increase when longer time periods are considered.

Importantly, opportunity costs are also likely to vary over time as the prices of commodity outputs and the inputs used to produce them vary. For example, higher output prices and/or lower input prices will increase profitability, thereby raising the opportunity cost of displacing agricultural production. Conversely, falling profitability leads to lower opportunity costs. This inevitably poses problems in setting payment rates equal to compensate for income forgone (Reed et al., this issue), but also incurs the risk that restoration gains may be undone if opportunity costs rise significantly in the future. For example, rising commodity market prices may cause voluntary participants in agri-environment schemes to revert to commodity production at the end of a scheme's contracting period (Colman, 1994; Whitby, 2000; Hodge and Reader, 2010).

Generally, guiding questions for choosing an appropriate timeframe for the analysis are: how sensitive are the outcomes of the analysis to the choice of the timeframe? For example, would different time periods favour particular restoration activities? Would longer or shorter time periods change conclusions about the relative merit of restoration activities compared to conservation of near-natural sites or maintenance of sites in their current condition?

These questions are also related to the chosen approach to discounting. The need for discounting arises because individuals are not indifferent with respect to the timing of costs and benefits<sup>6</sup>.

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<sup>6</sup> For further details on discounting and its implications, the reader is referred to, for example, Pearce et al. (2006), and, within a climate change context, Nordhaus (2008), Dasgupta (2008).

Individuals tend to prefer to obtain benefits earlier rather than later, and to defer costs. ‘Weights’ (discount factors) are therefore applied to costs and benefits accrued at different points in time. Discounted costs and benefits can then be expressed in a present value and combined to derive the Net Present Value (NPV), or to estimate their ratio (Benefit-Cost Ratio or BCR). Because restoration costs are often highest in the near future, whereas benefits take time to be generated, discounting may affect NPV and BCR in favour of doing nothing, or investing in alternatives with higher expected benefits in the short term. The choice of the timeframe for analysis can therefore have considerable implications on CBA outcomes, as does the choice of the discount rate. In a public policy context, a social discount rate should apply, which reflects social time preferences. Use of a (higher) private discount rate may be necessary to explain current and future land use choices by individual actors (see Bateman, 2009). When costs and benefits reach beyond one generation, there is a theoretical argument for using declining discount rates (e.g. Gollier et al., 2008). This would tend to increase the NPV of restoration activities with long term effects, as the effects in the future would contribute more to the NPV than when a constant discount rate is used.

#### 4.3 Spatial scale

It is important to consider that the ES vary in the spatial scale at which they provide benefits. For example, carbon sequestration has global benefits in terms of climate change mitigation, whereas recreational benefits accrue mainly to regional, and to a lesser extent (inter-)national, stakeholders. Understanding where the ES flows are produced across a landscape, and where the benefits are delivered can help to ensure that all benefits and costs are accounted for and internalised in the decision-making process. Knowledge on the distribution of costs and benefits across space helps to define the spatial boundaries of the CBA in defining who has standing in the CBA (see Step 1 in Figure 1).

The scale at which ecosystem services should be analysed from an economic point of view may differ from the boundaries of an ecosystem relevant to political or biophysical analyses (Bateman et al. 2006). Economic analyses of benefits tend to focus on the scale at which benefits are enjoyed, i.e. assess the benefits enjoyed at a populated unit. Economic analyses of supply-side measures (to increase ES production) and biophysical ES analyses tend to have a stronger focus on the location where measures will be implemented and how much of each ES flows from a spatial unit of peatland. For example, recreation benefits are enjoyed at the recreational site, but to understand how these benefits are distributed across stakeholders, we need to know where recreationists live and travel from. Similarly, water quality improvements may increase water quality downstream, hence the benefits are not enjoyed at the site itself and the economic analysis will have to focus on the preferences of downstream water users.

Differences in the spatial distribution of benefits and costs may imply that peatland restoration can have net positive benefits at the aggregate level of the society, but net losses on the individual level (e.g., for a farmer who can no longer maximise income from his/her land). Vice versa, it also implies that policy plans for peatland restoration that only consider local benefits may not provide results that are optimal at a global scale. Consideration of the spatial distribution of benefits of ecosystems may help to identify stakeholders and understand conflicting preferences over land use scenarios (see Reed et al., this issue, for a more detailed discussion of this point).

## **5 Spatially explicit ES assessment and valuation of peatland restoration**

Spatially explicit modelling of ES helps to reveal how benefits as well as management and opportunity costs vary across space and contexts, following spatial variation in biophysical characteristics (e.g., the condition of peatlands, climate, altitude) and socio-economic factors (e.g., population density and distribution, road networks, income, land ownership, land use). The development of spatially explicit ecological and economic models requires the compilation of a large geodataset, which may not be readily available and would likely have to be collated from various information sources.

### **5.1 Spatial variation in service production and delivery**

Biophysical delivery of services will vary for different peatlands across the UK, as a consequence of different environmental conditions, including morphological and climatic conditions and resulting differences in habitat structure. The difference between lowland and upland peatlands is a prime example. Additionally, delivery of services is dependent on the current condition of peat, which varies across space, and changes in service flows differ depending on the type of restoration practice implemented<sup>7</sup>. The placement of restoration activities within a peatland area can also affect ES delivery: for example, peatland restoration can result in reduced occurrence of high flash flows from a catchment (a proxy for flood risk), depending on the placement of the restoration activity within the catchment, and on environmental factors, such as the precipitation patterns across the catchment (Holden et al., 2004; Holden, 2005).

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<sup>7</sup> A detailed overview of the heterogeneity of peatland states, environmental factors determining state, habitat types and structures is given in JNCC (2011). Regarding service delivery, Worrall et al. (2010, 2011) show how delivery of carbon benefits may vary depending on changes in land management, and Worrall et al. (2011), Artz et al. (2012) and Birnie & Smyth (2013) report differences in emission factors and carbon budgets for different peatland types, management practices and peatland conditions. Quick et al. (2013) extend consideration to variation in other ecosystem services in the context of developing place-based restoration schemes. Other papers in this Special Issue also offer some discussion of the issues.

While the determination of the scale of analysis is informed by an understanding of the spatial variation in ES production and flow, and the associated costs and benefits, spatially explicit analysis of ES requires understanding of the spatial *units* (e.g., pixels of a certain size in a geographic information system) of peatland that can be analysed in terms of their ES flows and impacts of restoration. For these spatial units of analysis, the differences in ES flows over time arising from the implementation of a pre-defined set of restoration practices relative to the baseline have to be estimated. These flows depend on the spatial relationship between the spatial units of peatland, as outlined in Box 1.

In the most straightforward case, changes in ES flows resulting from the restoration of one spatial unit (e.g., pixel on a map) are independent of any changes in surrounding units of peatland (see example A, Box 1). Once biophysical changes for the range of ES under investigation have been established for each unit, they can be linked with information on benefits and costs arising from the change to estimate the net benefits of applying different restoration practices to the unit.

In most cases, however, the assumption of spatial independency of ES flows does not hold due to spatial interactions, connectivity between spatial units (e.g., upstream-downstream effects, spill-over effects), and non-linearity in ecological response with respect to area under restoration. For example, with respect to non-linear effects, Yang et al. (2010, 274) show that there is a non-linear relationship of service provision for peak flow reduction and sediment reduction following an increasing scope (area in ha) of wetland restoration in a Canadian prairie watershed (see examples B and C, Box 1). Non-linearity may take different functional forms, including exponential or (inverse) U-shaped relationships (i.e. there may be a positive contribution of adding more area to restoration up to a point where adding further restoration areas has a negative effect on ES provision).

Connectivity across peatland areas means that, due to biophysical linkages, restoration on any given land unit may influence neighbouring units and possibly more distant sites as well (Labadz et al., 2010). Connectivity may be constrained to those units under restoration, or those units that exhibit similar ecological conditions (see example C, Box 1). For example, restoration of an otherwise degraded peatland may improve habitat connectivity and therefore the living condition of certain plant or animal species (Bartelt et al., 2010).

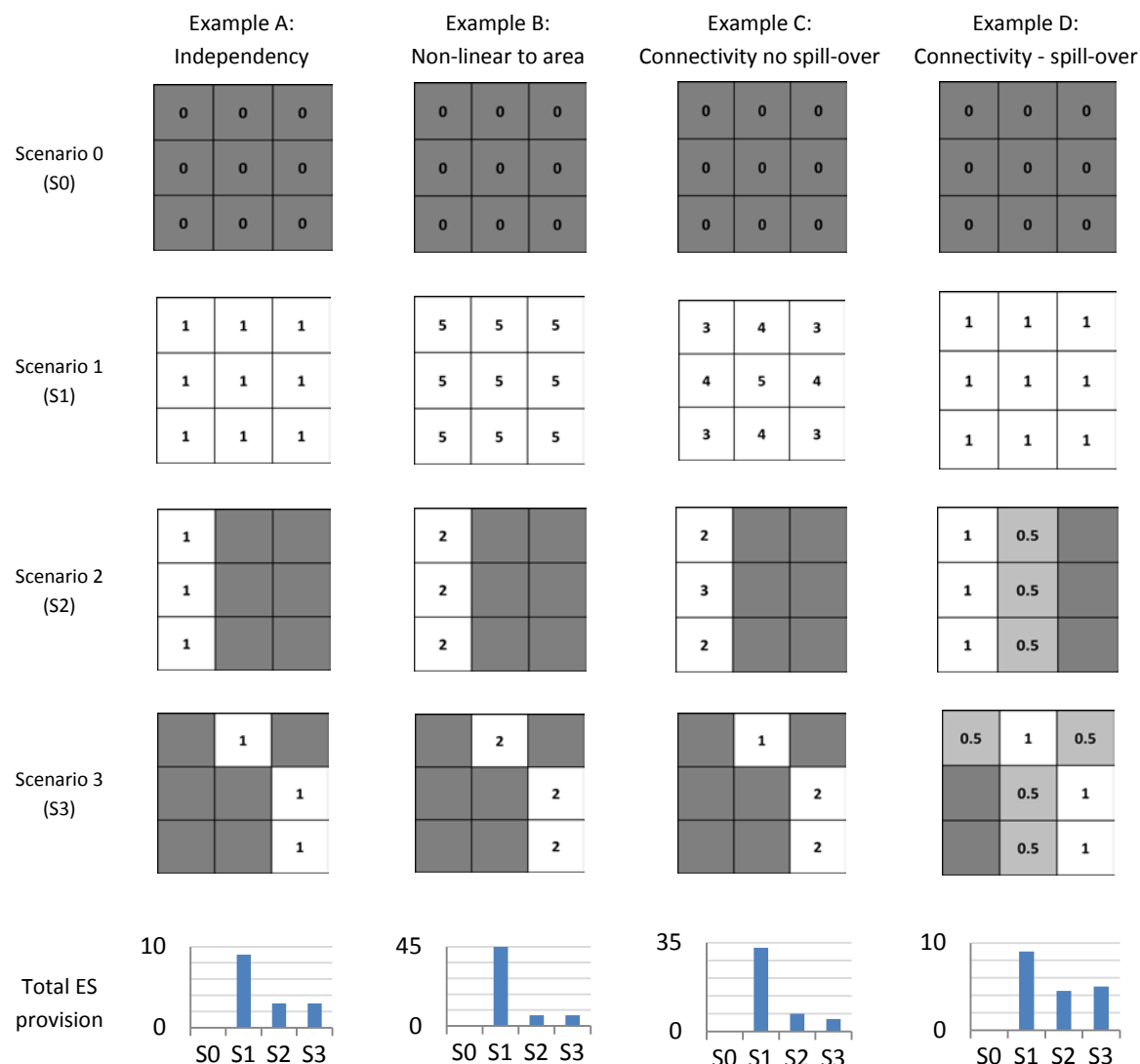
On the other hand, restoration activities may affect adjacent units that are not under restoration. Due to hydrological connectivity, for example, the total area of land affected both in terms of GHG emissions and other ES, including provisioning services from agricultural land use, may be larger than only the in-situ consequences. Such spill-over or ‘halo’ effects need to be taken into account for predictions of changes in ES flows, and assessments of opportunity costs of restoration activities (see example D, Box 1).

### Box 1 Biophysical ES delivery across space

Each 3x3 block represents a peatland ecosystem, with each pixel as the smallest spatial unit. Each pixel can take three 'states':

- Dark grey: degraded peatland, no restoration – ES provision=0;
- White: restored –ES provision>0;
- Light grey: partly improved –ES provision=0.5.

In the baseline scenario (S0), the peatland is fully degraded. The other three scenarios represent different restoration programmes. In scenario (S1), the peatland is fully restored, whilst in scenarios (S2) and (S3), only part of the peatland ecosystem (3 pixels) is restored.



#### Example A: independency

- For each ES, the provision level in each spatial unit (pixel) is independent of changes in other ES and in adjacent areas (pixels) or the wider ecosystem (block).
- The overall magnitude of changes in ES flows following restoration for a given area of peatland is equal to the sum of the effects of individual pixels.
- Marginal values of ES flow changes derived from restoration practices can be based on a fixed value per spatial unit
- ES provision: Scenario (S1)=9, Scenario (S2)=3, Scenario (S3)=3.

#### Example B: non-linear to area

- ES provision is subject to non-linear effects, where the effect on ES flow increases disproportionately to an increasing area under restoration
- No fixed value per pixel can be used to determine the values of changes in ES flows; it depends on the relationship between ES provision and area under restoration
- ES provision: Scenario (S1)=45, Scenario (2)=6, Scenario (3)=6

#### Example C: connectivity but no spill-over

- ES provision is subject to non-linear effects, where the ES flow increases disproportionately to an increasing area under restoration, but is also dependent on adjacent pixels under restoration.
- No fixed value per pixel can be used to determine values of ES flow changes: it depends on adjacency to other restored units
- ES provision: Scenario (S1)=33, Scenario (S2)=7, Scenario (S3)=5

#### Example D: connectivity with spill-over

- Spill-over ('halo') effects (pixels): due to biophysical linkages, restoring one pixel may influence neighbouring pixels that do not have to be under restoration.
- Spill-over effects may vary with the type and intensity of the restoration activity, and the current condition and land use in neighbouring peatland areas.
- ES flow changes in non-restored units adjacent to restored units need to be considered for valuation
- ES provision: Scenario (1)=9, Scenario (2)=4.5, Scenario (3)=5

The phenomena in the ecological responses to restoration activities described in Box 1 (examples B-D) are not mutually exclusive and may therefore occur simultaneously. They imply that changes in ES flows arising from restoration of a particular unit of peatland can often not be evaluated in isolation of adjacent or connected peatland areas and may prohibit generalising the findings from smaller spatial scales to whole catchments. For some ES, this can complicate a spatially explicit assessment of changes in service flows.

## 5.2 Monetary Valuation of peatland ES

### 5.2.1 *Marginal values of changes in ES flows*

Economists value marginal welfare changes associated with marginal changes in ES production/delivery. Marginal values can be related to a relatively small change in ES provision, or to the ES impact of a small increase in the area under restoration. The change should be considered marginal at the scale of the policy level of interest (Fisher et al., 2008), e.g. changes may be considered non-marginal at the local scale, yet marginal at the national scale – in which case economic valuation can still be meaningful at the national scale. For example, the complete loss of a certain peatland site may be considered marginal at a national scale if the area of that site is small relative to the total area of peatland within a country, but represents a non-marginal change at the local scale.

Marginal values for changes in ES flows do not have to be constant over provision levels or stocks (Bateman et al., 2011a). Indeed, values of ecosystem services should reflect the relative scarcity of their supply: the more of a service can be consumed or enjoyed, the less the value of consuming or enjoying an additional unit of that service. Marginal values associated with, for example, recreation and water supply/regulation may decline as the absolute magnitude of service flows and the associated stock of restored peatlands increase. On the other hand, the marginal value associated with further reductions in GHGs is not diminishing considerably with an increasing level of provision, because the total impact of restoring all of the UK's peatlands will be a relatively small contribution to mitigating climate change worldwide (in other words, the contribution is marginal at a global scale). The distinction between constant and diminishing marginal values matters, because constant marginal values can simply be multiplied with associated ES units to arrive at a total value to be used in CBA, while non-constant (diminishing) marginal values require integrating over ES provision levels for an interval reflecting the change in ES provision resulting from restoration.



### 5.2.2 Valuation techniques

There are various methods for valuing changes in ES flows, which have been well-documented in the literature (e.g. Bateman et al., 2002; Freeman, 2003). For carbon sequestration, methods that are commonly used include market-based pricing, for example using EU ETS carbon market information, cost-based prices (e.g., UK-DECC rates (DECC, 2009)), or estimates of the social cost of carbon (e.g., Tol 2005, 2010).<sup>8</sup> For non-use values, related to, for instance, biodiversity and cultural landscape values, for which no markets exist and for which we cannot draw on existing behaviour, survey-based Stated Preference (SP) methods can be used. In SP surveys, hypothetical markets are created and people are asked to state their Willingness-to-Pay (WTP) directly. Despite being well-established and widely adopted, the validity and reliability of these methods is, however, a topic of on-going debate (e.g., TEEB, 2010). Water regulation or flooding benefits can be valued via the damage cost avoided (by looking at the value of property that would be damaged would a flood occur), and again by SP studies (which can also take into account the emotional damage of losing one's house or personal items, or the discomfort of evacuations). Water supply regulation could be valued using the market price of water (with the caveat that these prices may be distorted by subsidies), or by assessing the costs that water companies would have to bear if water quality or supply from the ecosystem changed. Changes in water treatment costs resulting from water quality changes are a lower bound WTP estimate for consumer demand for a certain quality of drinking water.

For recreational benefits, travel cost methods are widely used (Freeman, 2003). In case of peatland restoration, the question here would be to understand how demand for recreation varies for differently managed peatlands. For instance, accessibility may differ across land management regimes, but also the wildlife that attracts people may differ. SP surveys can also be used to assess the recreational use-value of peatlands and benefits of game.

The private benefits of agriculture and pasture are usually assessed using market prices. The main question is, how much the output of farmers changes when peatland management changes, or when external factors such as world market prices for inputs and outputs change (Hanley et al., 2012). Peatland may provide a number of raw materials, such as peat for energy, soil for horticulture, withies and timber. These can be valued using market prices, although care has to be taken to account for price distortions arising from market controls (e.g. import tariffs) and transfer payments via subsidies (e.g. farm payments).

Although there are a few valuation studies that specifically focus on peatlands (see Wichmann et al., in press), studies on moorlands and heathlands (which have peaty top layers) exist, including for the UK. The UK National Ecosystem Assessment has combined these types with mountain ecosystems

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<sup>8</sup> In general, market prices and cost-based estimates (e.g. DECC rates, which are based on the abatement cost method, are a second-best option as they do not include the Consumer Surplus as part of social welfare.

(see Tinch et al., 2010). Tinch et al. (ibid) summarise relevant studies for these areas. For example, White and Lovett (1999) assess the WTP for nature conservation in the North York Moors NP using Contingent Valuation. Black et al. (2010) use Contingent Valuation to assess public preferences for landscape and wildlife in the North Pennines. Hanley et al (2007) apply choice modelling to a range of landscape types, including heather moorland and bog. Christie et al (2011) used choice experiments to value Biodiversity Action Plans, including in relation to peatland degradation and restoration. The patchiness and relative lack of literature on economic values of peatland ecosystems in the UK suggests that there is a need for collecting additional primary valuation data to enable the CBA approach to spatial targeting outlined in this paper.

### *5.2.3 Spatial heterogeneity in economic values of ES benefits*

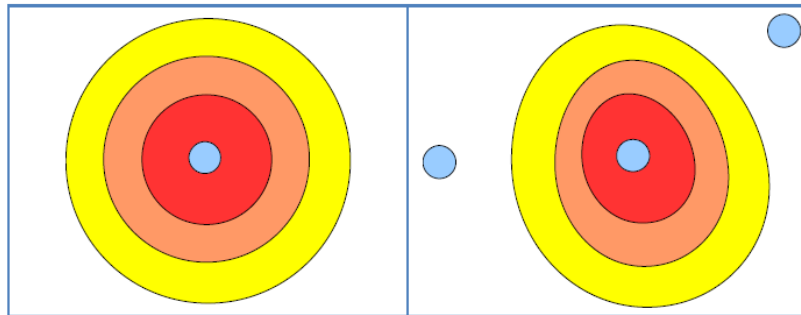
As Colombo and Hanley (2008) show, marginal WTP for changes in the area under heather moorland and bog varies across regions. In general, differences in people's socio-economic, spatial and biophysical context may affect preferences and therefore result in spatial heterogeneity in WTP. The value of a hectare of restored peatland will vary across the country due to differences in local preferences, local scarcity, and local socio-economic characteristics. There are a number of spatial patterns in economic values, where values are related to distance and accessibility, the availability of substitutes, and directional effects.

Distance-decay is the term that refers to a decline in WTP as distance to the site where ES are delivered increases. Distance-decay has a theoretical basis in travel cost studies, as the cost of recreation will increase (and demand will decrease) as distance increases. Distance may also serve as a proxy for other determinants of WTP that vary across space, including familiarity and information. Non-use values may not have a direct link with distance. Empirical studies on distance-decay in non-use values of environmental goods and services have found mixed results, however, and some find significant distance-decay effects in non-use values (Hanley et al., 2003; Bateman et al., 2006; Jørgensen et al., 2012). Distance-decay rates may vary between ES and populations – there is no one-rate-fits-all estimate.

The main reason to assess distance-decay is to delineate the economic market or jurisdiction, i.e. the area over which the population has a positive willingness to pay for a change in ES provision obtained from a particular ecosystem. WTP estimates per individual or household can then be aggregated over the population living in this economic market area to estimate the total WTP held by society.

Although economists have often modelled space in a one-dimensional setting, distance-decay rates may vary across directions, either because the flow of ES has a prevailing direction, or because there are differences in the distribution of substitutes around the study site (Schaafsma et al., 2012), or

because there are differences in place attachment across areas (Brouwer et al., 2010). Substitution effects take place when people have alternative site providing similar ES that may be substitutable to some degree; the higher the supply of alternatives, the lower the WTP for the study site. Compare the two situations in Figure 3: left is a situation without substitutes (uni-directional distance-decay), right is a situation with substitutes and therefore stronger distance-decay in the direction of the substitute (multi-directional distance-decay).



**Figure 3** Distance decay without (left) and with (right) substitution effects. Source: Schaafsma (2010). Note: blue circles represent sites providing ES, surrounding circles reflect per household WTP values subject to distance-decay: red are high WTP values, yellow are low WTP values.

Directional effects (i.e. variation in individual WTP estimates across directions compared to the point where ES are produced) when (a) the flow of ES has a certain direction (e.g. downstream water quality effects, downwind air pollution effects), (b) when the substitutes are not randomly distributed across space, or (c) when there are particular differences in unobserved variables affecting WTP that vary across directions. Ideally, one would capture these effects in the WTP model by including the relevant variables that drive WTP, rather than using distance as a proxy for these underlying factors. Alternatively, directional heterogeneity in space can be modelled by using directional dummy variables (Schaafsma et al., 2012), (more continuous) spatial trend variables (Cameron, 2006; Schaafsma et al., 2013), or regional dummy variables (Brouwer et al., 2010).

#### 5.2.4 Spatial variation in (opportunity) costs of restoration

In lowland areas, peatland restoration may displace relatively intensive and profitable horticulture or agricultural cropping - implying a high (financial) opportunity cost. By contrast, extensive upland agriculture or forestry is typically less profitable and thus opportunity costs are likely to be lower. Indeed, such spatial variation is already acknowledged in differentiated payment rates for income foregone<sup>9</sup> under peatland-related agri-environment schemes in different parts of the UK (Keenleyside

<sup>9</sup> Payments may also include elements of additional costs incurred, but these are not identified explicitly.

and Moxey, 2011). For example, peatland-related options under the High Level Scheme in England offer around £40/ha for upland sites and £150 for lowland sites (Natural England, 2010a).

However, such crude differentiation masks likely heterogeneity of opportunity costs across sites. For example, not all upland peatland sites will be equally productive in agricultural terms or in the degree of actual displacement incurred - differences in biophysical conditions, farm size, access to capital and/or management skills mean that farms are likely differ greatly in their flexibility<sup>10</sup> to (re)allocate resources and (re)locate a particular enterprise to reduce their actual cost incurred. Hence in some cases opportunity costs may be significant whilst in others they may be close to zero. Such variation is generally unobserved. Where opportunity costs for peatland restoration have been estimated, they are generally based on simulation modelling (Cortus et al., 2011; Armsworth et al., 2012). Neglecting to design payment for ecosystem service schemes which ignore this spatial variation in opportunity costs has been shown to impair policy cost-effectiveness (Armsworth et al., 2012).

Opportunity costs are also affected by future changes in climatic conditions and policy developments, for example with respect to the EU Common Agricultural Policy, which in turn affect the spatial allocation of land use. For example, climate change may generate potential agricultural benefits in the northern half of the UK, while the area under permanent grassland and rough grazing is expected to decline (Fezzi et al., 2011). Spatially targeting areas for peatland restoration hence requires the use of scenarios where these effects are outlined.

Capital costs and on-going management costs are also likely to vary. In general, the greater the degree of degradation, the higher the costs, but factors such as scale of restoration and ease of access to the site may also exert an influence. Capital costs arise from items such as fencing for grazing control, plastic piling or heather bales for blocking ditches, and plants or seeds for re-vegetating bare peat. Unit costs for such items may vary slightly across sites (and over time), but total capital costs at a given site will be driven primarily by how much of each item is required according to site conditions (Lunt et al., 2010; Tanneberger and Wichtmann, 2011).

Management costs will be incurred through the effort of undertaking initial capital works – for example in blocking ditches – but also in capital maintenance and other activities that may arise periodically over time such as re-profiling gullies or controlling scrub encroachment. In addition, some monitoring effort will be required to gauge progress and to inform on-going management. Again, unit costs of particular activities may vary slightly between sites and over time, but total management costs at a given site will reflect the aggregate effort involved and will thus depend more on the volume, intensity and duration of management activities (Lunt et al., 2010; Tanneberger and Wichtmann, 2011).

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<sup>10</sup> Indeed retaining flexibility over management of their land may represent an option value that increases the opportunity cost of change beyond simply the apparent income foregone from a given parcel of land (Behan et al., 2006).

Some estimates of capital and management costs are contained within various attempts to compile and assess information on restoration projects (e.g. Defra, 2008; Natural England, 2010a&b; Bonn et al, 2011). Harlow et al. (2012) use local stakeholders and expert knowledge to estimate costs for a specific catchment area whilst Moran et al. (2013) use values reported in the literature to set plausible ranges for different cost elements that can then be subjected to crude sensitivity analysis for individual sites. Reported costs vary from a few hundred pounds per hectare to several thousand. However, within this, capital and management costs are rarely reported separately and monitoring costs seldom noted at all. In many cases, lack of cost data reflects the relative newness of projects where capital works have only been completed recently and on-going management has yet to be undertaken.

However, more general measurement problems also hinder the reporting of costs. In particular, sites vary considerably in their characteristics - for example in terms of the size and density of drainage grips or the proportion of bare peat present. Not only does this mean that different restoration activities will be required for different sites but also that the volume and intensity of a given activity and/or the combination of activities will vary (Lunt et al., 2010; Tanneberger & Wichtmann, 2011). For example, a lightly degraded site may require only limited grip blocking whilst a more heavily degraded site may require more extensive grip blocking but also stabilisation and re-vegetation of bare peat. Equally, afforested sites will require tree removal.

In addition, attempts to collate results across different projects are hampered further by differences in how costs are reported, including problems in defining the areal extent. That is, hydrological connectivity between parcels of land means that restoration activities at a specific location will typically exert an influence on adjoining parcels of land plus possibly some further away. This means that specifying the number of hectares (and by extension the volume of ecosystem services affected) becomes awkward and thus expressing definitive costs per hectare somewhat difficult. Development of the Peatland Carbon Code includes efforts to refine cost data (Defra, pers. comm. 2014).

## **6 Risk and uncertainty**

There are many sources of uncertainty that may influence cost and benefit estimates derived from restoration of peatlands. The basic approach to considering these is to investigate the influence of changes in uncertain factors and assumptions on outcomes in a sensitivity analysis (Step 4 in Figure 1). For example, predictions about future states of peatlands and the effectiveness of restoration activities with respect to ES delivery are sensitive to assumptions about climate change over time. Model estimates (both regarding ES impacts and regarding the estimation of benefits) are subject to error, which can usually be described in terms of upper and lower bounds of confidence. The choice of the timeframe and discount rate can influence results.

The brief discussion in this section focuses on risk and uncertainty related to projected outcomes, and how information on outcome-related risk can be incorporated into environmental valuation. Information on the probability of facing a certain outcome can be incorporated into a quantitative assessment of costs and benefits, if there is some information about the probability distribution of future projections and related ES impacts. Therefore, it would be useful to identify, for each source of outcome-related ‘uncertainty’, whether it can be described in probabilistic terms as an element of risk in the sense of Knight (1921), or not, in which case it represents a source of ‘true’ uncertainty. Furthermore, it would be useful to understand if outcomes are risky, because they depend on stochastic events (e.g., weather patterns), or because of data and modelling limitations. The first constitutes an inherent risk that cannot be reduced, while the latter could be reduced by expending research time and effort on improving the scientific database and models.

For example, Worrall et al. (2010) present a probabilistic model of achieving net GHG benefits from different land use interventions on peatlands, where model uncertainty is caused by a lack of data. Their Bayesian model can be updated and uncertainty reduced, once additional data from field studies becomes available. With an increasing amount of data from field studies over time, it will be possible to improve the understanding of what explains variation in estimates of GHG emissions from different interventions. This may in turn help to better predict how local conditions affect the effectiveness of restoration with respect to GHG emission reductions.

The analysis of Worrall et al. (2010) shows that, given the current limited amount of data, some land use interventions have a non-zero probability of actually turning peatlands into a net source of emissions. Similarly, there is a risk that peatland restoration may actually result in enhancing peak flow events rather than mitigating them (Holden, 2005). Such findings clearly illustrate the importance of taking risk related to the delivery of ES outcomes into account. It also demonstrates the potential usefulness of identifying factors influencing variability, for example with respect to determining the local conditions that increase or decrease the likelihood of a site to become a net source of GHG emissions after restoration.

In the presence of risk, restoration outcomes in terms of ES can be weighted by their probabilities to occur and related to their respective benefit estimates to derive an expected value of a given change. This approach assumes risk neutrality of the individuals over the range of different outcomes. However, people tend to be risk averse, implying that they would demand a risk premium for more risky restoration alternatives. Risky alternatives may therefore be penalised ex-post valuation based on different assumptions about risk aversion made by analysts. Alternatively, information on outcome-related risk may also be incorporated directly into valuation methods (Glenk and Colombo, 2011; Roberts et al., 2008; Rigby et al., 2010). However, this is of not much help when possible states of the world or their probabilities are unknown.

## 7 Key drivers of restoration

In this section, the impacts of restoration on climate change-related benefits and on biodiversity are outlined in greater detail. Both impacts contribute significantly to the increasing interest of policy makers and conservation NGOs in peatland restoration, and are therefore important elements of an ES based assessment of peatland restoration. The impacts of restoration on water-related services and the relevance of peatland restoration to compliance with the EC water Framework Directive (WFD) have recently received greater attention and are covered in a separate article in this special issue (Martin Ortega et al., this issue). A key policy instrument that can stimulate or hamper land use change on farmed peatlands within the EU is the Common Agricultural Policy (CAP) and its Pillar 2 instruments, which may offer opportunities for providing financial incentives for peatland restoration to landowners. For a focussed discussion of policy drivers and barriers to implementing peatland restoration, see Reed et al. (this issue) and Bonn et al. (this issue).

### 7.1 Climate change-related benefits

Given the ability of peatlands to sequester and retain atmospheric carbon and the high emission levels associated with degradation of peat, management and restoration of peatlands merit serious consideration as climate change mitigation activities (Bain et al., 2011; Bonn et al., this issue), and accounting for GHG emissions from wetland rewetting, including peatland restoration, has become an option that Parties to the United Nations Framework Convention on Climate Change can opt into if they wish (Article 3.4 of the Kyoto Protocol). At national level, peatland restoration can contribute to achieving national emission reduction targets. For example, the Climate Change (Scotland) Act (2009) requires emission reductions of 80% from 1990 levels by 2050; if it can be demonstrated that damage to peatland pre-dates 1990, restoration and associated emission reductions can count against this target. Although subject to a number of caveats, illustrative figures for peatland management and restoration (Natural England, 2010a, 2010b; Morris et al., 2010; Moxey, 2011; Harlow et al., 2012; Graves and Morris, 2013; Moran et al., 2013; Pettinotti, 2013) suggest that their cost-effectiveness as GHG mitigation options compares favourably to other mitigation options already receiving some policy support (Bonn et al., this issue). As such, neglecting peatland management and restoration places a greater burden on other, more costly, mitigation options and reduces the overall mitigation capacity available to meet challenging emission targets.

Despite some uncertainty over the GHG emission reduction potential of peatland restoration (e.g., Worrall et al., 2010), there is a sufficient scientific basis for the long-term net GHG benefits of restoration compared to inaction in degraded peatlands (Bain et al., 2011; Bonn et al., this issue). Prices per tonne of CO<sub>2</sub> equivalent are fixed across space, but the costs and co-benefits of mitigating CO<sub>2</sub> emissions from peatland are spatially heterogeneous. Hence, if climate change mitigation becomes the principal policy driver for restoration, the development and analysis of scenarios of change could be simplified by first focusing on a spatially explicit assessment of cost-effective

mitigation of GHG emissions. Effects on ES such as water regulation or recreation could be part of a second step of analysis: impacts on additional ES would only be assessed for a single scenario that represents the most cost-effective allocation of restoration activities with respect to GHG emission reductions. In the absence of valuation data on such co-benefits, expert judgement may be used to adjust the initial ranking of priority sites derived from the cost-effectiveness analysis of restoration activities for climate change mitigation (Reed et al., this issue).

## 7.2 Biodiversity

Biodiversity conservation can motivate public and private investments in peatland restoration, given the high proportion of peatland sites that are designated for their conservation significance under national and international law. Key policy drivers for peatland restoration are at global level the Nagoya protocol (Aichi targets) of the UN Convention on Biological Diversity and the Ramsar Convention, at EU level the Biodiversity Strategy, the EU Bird Directive and the EU Habitats Directive. In the UK, biodiversity policy is implemented through the UK Biodiversity Action Plan (UK BAP 1994). Indeed a main recent driver for restoration in the UK has been to achieve favourable condition for Sites of Special Scientific Interest (SSSI) (Worrall et al. 2011).

Numerous studies based on stated preferences indicate that biodiversity is valuable (e.g., Christie et al., 2006; Czajkowski et al, 2009; Christie and Rayment, 2012; Jobstvogt et al, 2014) and that biodiversity conservation is a desirable goal. People also reveal their preferences for biodiversity conservation by making charitable payments to environmental NGOs such as RSPB, indicating that there is a WTP for biodiversity conservation. This subsection is concerned with the non-use aspects of biodiversity and those benefits from biodiversity that are not related to the supporting role of biodiversity for providing ES and are hence already captured in the value of other ecosystem-services.

In the context of peatland restoration, there is a need to better understand which dimensions of biodiversity people value (see Christie et al., 2006, for different biodiversity concepts). Value may be derived from the knowledge and experience of a greater number of a certain species, a greater ‘diversity’ of certain types of plants or animals (e.g., birds, flowering plants). Economic value following from people’s preferences is found to be mainly related to key charismatic species that are visually attractive (such as birds), but charismatic species may not be those which are most crucial to the ecological functioning of the system, hence not have the highest ecological value. Value has also been related to (descriptions of) rarity (see Samples et al., 1986), although ‘rarity’ is context dependent.

Stated preference studies have been used to place monetary values on biodiversity, but the reliability of biodiversity studies that focus on non-use values is widely debated. The UK National Ecosystem



Assessment concluded that results from such valuation studies, while clearly indicating that biodiversity is valuable (i.e. that  $WTP > 0$ ), should be treated with some caution in terms of exactly what they are measuring (see Bateman et al., 2011b, 1081-1086).

An alternative way of incorporating biodiversity objectives into a framework aimed at spatial targeting of peatland restoration is to apply a ‘no-regrets approach’ to the selection of restoration measures at given peatland sites. Such an approach limits the possibilities of trade-offs between biodiversity and other ES, especially commercialised provisioning services (e.g. livestock, timber or grouse production), and implies that a specific area and restoration activity is only considered if applying the management change would not compromise biodiversity conservation objectives. These biodiversity conservation objectives can be based on existing policy guidelines, strategy documents and targets such as the UK Biodiversity Action Plan. Alternatively, a set of objectives could be developed that reflect a broad consensus across society. Mace et al. (2011) outline common conservation perspectives and discuss them in the light of an ecosystem services approach, highlighting the difficulties and challenges to accommodate pre-dominant conservation paradigms in an ecosystem services framework. Ultimately, setting conservation objectives involves value judgments and objectives need to be scrutinised with respect to their social and political acceptance. Given the heterogeneity of (sometimes strong and conflicting) views across society, it may be difficult to easily establish a consensual perspective.

## **8 Possible approaches to reduce the complexity of the analysis**

To derive an allocation of peatland restoration activities across peatland sites that maximises social welfare, knowledge on the ecological response to restoration activities is required for all possible combinations of restoration practices and peatland units to be restored. Limiting the number of different restoration practices and intensities considered at particular peatland sites across the country, for example via expert-based pre-selection, can significantly reduce the complexity of the analysis. Other simplifying criteria or rules could be introduced. The definition of a small set of scenarios that are evaluated in terms of ES impacts and associated costs and benefits can reduce the need to optimise across a large number of factors and locations. For example, targets for restoring and conserving a certain proportion of peatland area could be set, and additional indicators (socio-economic, ecological, geographical distribution, ‘risk’ of severe degradation, level of outcome related risk) could be used to inform a reduced set of scenarios for evaluation.

Biophysical response functions of ES to management change can be non-linear and exhibit complex spatial relationships, and are likely to vary across different peatland sites. The establishment of generally applicable biophysical response functions that vary as a function of restoration practices and a number of environmental factors would facilitate the analysis of biophysical ES impacts (see Evans

et al., this issue). The presence of independent ecological response units, i.e. where ES provision from a subset of spatial units is similar and independent of other units, would further simplify the spatially explicit analysis.

Because marginal cost and benefit functions can be non-linear, too, the relationship between the monetary value and the biophysical unit needs to be understood. Again, this relationship may vary across peatland sites. Therefore, primary data on benefits and opportunity costs should ideally be collected for a range of peatland sites across the country. When this is not possible, benefit estimates can be inferred from previous valuation studies in similar contexts (see Evans et al, this issue; and Martin-Ortega et al., this issue for critical discussions of such an approach), although the lack of studies to draw on for such an exercise is limited and caveats apply to extend the database by, for example, including valuation studies on wetlands in general (Evans et al., this issue). Opportunity cost estimates can (continue to) be based on simulation instead of models using primary data.

Undertaking the analysis proposed in this paper will require a significant interdisciplinary research effort to both collect and collate the basic spatial information needed, to establish a sound scientific knowledge base for ES impacts in the presence and absence of changes in peatland management, and to relate these impacts to costs and benefits using appropriate (primary) valuation methods. Until the information necessary to conduct the analysis becomes available, simplifying assumptions that reduce complexity, but aim to minimise the risk of severely misrepresenting the ecological response and therefore ecosystem service delivery, may be necessary to provide practical guidance on policy-makers. The ‘no regret approach’ and the use of cost-effectiveness analysis discussed in sections 7.1 and 7.2 provide additional practical ‘short-cuts’ in this respect.

## **9 Conclusions and outlook**

Meeting the aim of spatially targeting peatland restoration from an economic perspective is a very demanding task, requiring close interdisciplinary collaboration. It constitutes a big opportunity to learn and to progress research by improving the biophysical understanding of peatland ecosystems as a basis for increasing and refining the evidence base on the social costs and values of peatland restoration. A major methodological challenge is to model the complex spatial and temporal effects in ecological responses to management change to assess biophysical ES delivery. Accounting for the spatial heterogeneity of preferences for benefit estimation, and deriving spatially explicit estimates of opportunity cost of peatland restoration represent further challenges to a spatially explicit CBA of peatland restoration. It is clear that the net social benefits of restoration will vary across space as opportunity costs, ecological and economic benefits are all spatially heterogeneous.

Aspects of risk and uncertainty are important and can be found at all stages of the analysis and in the application of both biophysical and economic models. Because uncertainty related to the delivery of ES outcomes can have large consequences on estimates of net present value for management changes at all spatial scales, aspects of risk and uncertainty should be given greater attention and be considered from the outset, for example by incorporating outcome-related risk directly in valuation methods.

The CBA approach to spatial targeting of restoration activities outlined in this paper can contribute significantly to the understanding of how peatland ecosystems contribute to social welfare. Specifically, it can inform how (public) funds should be used to maximise the contributions of restoration to social welfare, it can be used to evaluate suggested restoration practices, and to understand ES trade-offs across different peatland sites. This has implications for policy design in terms of how land is enrolled in restoration schemes (e.g. voluntarily, through specific invitation or general open tendering, or through some spatially-disaggregated PES scheme), how areas for restoration are targeted, and how payments are made (e.g. tied to compliance mechanisms or as additional sums).

However, given the significant research needs and the increasing policy interest to quickly progress peatland restoration, the question may be asked if the research effort necessary for developing a spatial targeting approach under consideration of a wide range of benefits derived from changes in ES flows would be justified by the expected gains in net social benefit. The answer to this question depends, in part, on the likely scope of government commitment to support management interventions on peatlands; and the social welfare implications of allocating resources in a sub-optimal manner. Key influences on this include the degree to which property rights are adjusted to extend the polluter pays principle more fully to rural land uses and how implementation of the latest reforms of the CAP aligns with restoration activities – in particular whether restoration is supported under Pillar II but also whether restored land remains eligible for Pillar I support.

Yet, as documented by Armsworth et al. (2012) in a study on cost-effective policies aimed at biodiversity improvements, simplifications in policy design can be very costly (especially where there is significant spatial variability in opportunity costs and ecological potential), and a spatially explicit approach to implementation can result in large efficiency gains. Cautiously transferring this finding to the case of a spatially explicit analysis of peatland restoration activities suggests that the benefits in terms of improved decision making on restoration are likely to outweigh the additional cost associated with such an analysis.

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## References

- Ács, S., Hanley, N., Dallimer, M., Robertson, P., Wilson, P., Gaston, K., Armsworth, P.R., 2010. The effect of decoupling on marginal agricultural systems: implications for farm incomes, land use and upland ecology. *Land Use Pol.* 27(2), 550–563.
- ASC, 2013. Managing the land in a changing climate – Adaptation Sub-Committee progress report 2013. Adaptation Sub-Committee, London. <http://www.theccc.org.uk/publication/managing-the-land-in-a-changing-climate/> (accessed 14 February 2014)
- Armsworth, P., Acs, S., Dallimer, M., Gaston, K., Hanley, N., Wilson, P., 2012. The cost of policy simplification in conservation incentive programs. *Ecol. Lett.* doi: 10.1111/j.1461-0248.2012.01747.x
- Artz, R., Chapman, S., Donnelly, D., Matthews, R., 2012. Potential abatement from Peatland Restoration., James Hutton Institute for ClimateXchange. Enquiry number 1202-02. <http://www.climateexchange.org.uk/index.php/cds/59-potential-emissions-abatement-from-peatland-restoration> (accessed 22 November 2012)
- Bain, C.G., Bonn, A., Stoneman, R., Chapman, S., Coupar, A., Evans, M., Gearey, B., Howat, M., Joosten, H., Keenleyside, C., Labadz, J., Lindsay, R., Littlewood, N., Lunt, P., Miller, C.J., Moxey, A., Orr, H., Reed, M., Smith, P., Swales, V., Thompson, D.B.A., Thompson, P.S., Van de Noort, R., Wilson, J.D., Worrall, F., 2011. IUCN UK Commission of Inquiry on Peatlands. IUCN UK. Peatland Programme, Edinburgh.
- Balmford, A., Rodrigues, A.S.L., Green, R.E., Fisher, B., Naidoo, R., Strassburg, B., Turner, R.K., 2011. Bringing ecosystem services into the real world: an operational framework for assessing the economic consequences of losing wild nature. *Environ. Resource Econ.* 48(2), 161–175.

- Bartelt, P E.; Klaver, R.W., Porter, W.P., 2010. Modeling amphibian energetics, habitat suitability, and movements of western toads, *Anaxyrus* (=Bufo) boreas, across present and future landscapes. *Ecol. Model.* 221(22), 2675–2686.
- Bateman, I.J., Carson, R., Day, B., Hanemann, W.M., Hanley, N., Hett, T., Jones-Lee, M., Loomes, G., Mourato, S., Ozdemiroglu, E., Pearce, D.W., Sugden, R., Swanson, J., 2002. *Economic valuation with stated preference techniques: a manual*, Edward Elgar, Cheltenham, UK.
- Bateman I.J., Lovett A.A., Brainard J.S., 2003. *Applied environmental economics: a GIS approach to cost-benefit analysis*, Cambridge University Press, Cambridge.
- Bateman, I., B. Day, S. Georgiou, Lake, I., 2006. The aggregation of environmental benefit values: welfare measures, distance decay and total WTP. *Ecol. Econ.* 60, 450–460.
- Bateman, I.J., 2009. Bringing the real world into economic analyses of land use value: incorporating spatial complexity. *Land Use Pol.* 26, S30–S42.
- Bateman, I.J., Mace, G.M., Fezzi, C., Atkinson, G., Turner, K., 2011a. Economic analysis for ecosystem service assessments. *Environ. Resource Econ.* 48, 177–218.
- Bateman, I., Abson, D., Beaumont, N., Darnell, A., et al. 2011b. *Economic Values from Ecosystems* [chapter 22], in: UK National Ecosystem Assessment. *Understanding nature's value to society*. Technical Report. UNEP-WCMC, Cambridge, 1067–1152.
- Birnie, R.V., Smyth, M.A., 2013. Case Study on developing the market for carbon storage and sequestration by peatlands. Crichton Carbon Centre.
- Behan, J., McQuinn, K., Roche, M., 2006. Rural Land Use: Traditional Agriculture or Forestry? *Land Econ.* 82(1), 112–123.
- Bonn, A., Holden, J., Parnell, M., Worrall, F., Chapman, P.J., Evans, C.D., Termansen, M., Beharry-Borg, N., Acreman, M.C., Rowe, E., Emmett, B., Tsuchiya, A., 2010. *Ecosystem Services of Peat - Phase 1*. Defra Project Code SP0572.
- Bonn, A., Reed, M.S., Evans, C., Joosten, H., Bain, C., Farmer, J., Emmer, I., Couwenberg, J., Moxey, A., Artz, R., Tanneberger, F., von Unger, M., Smyth, M.-A., Birnie, D., this issue. Investing in nature: developing a peatland carbon code. *Ecosystem Services* (submitted)
- Brouwer, R., Martin-Ortega, J., Berbel, J., 2010. Spatial preference heterogeneity: A Choice Experiment. *Land Econ.* 86(3), 552–568.
- Cameron, T.A., 2006. Directional heterogeneity in distance profiles in hedonic property value models. *J. Environ. Econ.Manag.* 51(1), 26–45.
- Carpenter, S.R., Ludwig, D., Brock, W.A., 1999. Management of eutrophication for lakes subject to potentially irreversible change. *Ecol. Appl.* 9, 751–771.

- Christie, M., Hanley, N., Warren, J., Murphy, K., Wright, R., Hyde, T., 2006. Valuing the diversity of biodiversity. *Ecol. Econ.* 58(2), 304–317.
- Christie, M., Rayment, M., 2012. An economic assessment of the ecosystem service benefits derived from the SSSI biodiversity conservation policy in England and Wales. *Ecosys. Serv.* 1, 70–84.
- Christie, M., Hyde, T., Cooper, R., Fazey, I., Dennis, P., Warren, J., Colombo, S., Hanley, N., 2011. Economic valuation of the benefits of ecosystem services delivered by the UK Biodiversity Action Plan. Report to Defra, London.
- Colman, D., 1994. Ethics and externalities: agricultural stewardship and other behaviour. *J. Agr. Econ.* 45 (3), 299–311.
- Colombo S., Hanley, N., 2008. How can we reduce the errors from benefits transfer? An investigation using the Choice Experiment method. *Land Econ.* 84(1), 128–147.
- Cooper, D.A., Evans, C.D., Zielinski, P., Levy, P.E., Gray, A., Peacock, M., Fenner, N., Freeman, C., (under review). Infilled ditches are hotspots of landscape methane flux following peatland restoration.
- Cortus, B., Jeffrey, S., Unterschultz, J.R., Boxall, P.C., 2011. The economics of wetland drainage and retention in Saskatchewan. *Can. J. Agr. Econ.* 59, 109–126.
- Couwenberg, J., Thiele, A., Tanneberger, F., Augustin, J., Bärtsch, S., Dubovik, D., Liashchinskaya, N., Michaelis, D., Minke, M., Skuratovich, A., Joosten, H., 2011. Assessing greenhouse gas emissions from peatlands using vegetation as a proxy. *Hydrobiologia* 674, 67–89.
- Couwenberg, J., Fritz, C., 2012. Towards developing IPCC methane ‘emission factors’ for peatlands (organic soils). *Mires Peat* 10, Article 03, 1–17.
- Czajkowski, M., Busko-Briggs, M., Hanley, N., 2009. Valuing changes in forest biodiversity. *Ecol. Econ.* 68, 2910–2917.
- Dasgupta, P., 2008. Discounting climate change. *J Risk Uncertainty* 37(2-3), 141–169.
- D’Astous, A., Poulin, M., Aubin, I., Rochefort, L., 2013. Using functional diversity as an indicator of restoration success of a cut-over bog. *Ecol. Eng.* 61B, 519–526.
- DECC, 2009. Carbon Appraisal in UK Policy Appraisal: A revised Approach. Climate Change Economics, Department of Energy and Climate Change.
- Defra, 2009. Safeguarding Our Soils. A Strategy for England. <http://archive.defra.gov.uk/environment/quality/land/soil/documents/soil-strategy.pdf> (accessed 13 February 2014).
- Evans, C., Bonn, A., Holden, J., Reed, M., Evans, M., Worrall, F., Parnell, M., this issue. Relationships between anthropogenic pressures and ecosystem functions in UK blanket bogs: Linking process understanding to ecosystem service valuation. *Ecosystem Services* (submitted).

- Fezzi, C., Crowe, A., Abson, D., Bateman, I., Askew, T., Munday, P., Pascual, U., Sen, A., Darnell, A., Haines-Young, R., 2011. Evaluating provisioning ecosystem service values: a scenario analysis for the United Kingdom. Report to the Economics Team of the UK National Ecosystem Assessment. <http://uknea.unep-wcmc.org/LinkClick.aspx?fileticket=h2WCf1fOqNk%3D&tabid=82> (accessed 13 February 2014).
- Fisher, B., Turner, K., Zylstra, M., Brouwer, R., De Groot, R., Farber, S., Ferraro, P., Green, R., Hadley, D., Harlow, J., Jefferiss, P., Kirkby, C., Morling, P., Mowatt, S., Naidoo, R., Paavola, J., Strassburg, B., Yu, D., Balmford, A., 2008. Ecosystem Services and Economic Theory: Integration for Policy-Relevant Research. *Ecol. Appl.* 18(8), 2050–2067.
- Fisher, B., Turner, R.K., Morling, P., 2009. Defining and classifying ecosystem services for decision making. *Ecol. Econ.* 68(3), 643–653.
- Fisher, B., Turner, R.K., Burgess, N.D., Swetnam, R.D., Green, J., Green, R.E., Kajembe, G., Kulindwa, K., Lewis, S.S., Marchant, R., Marshall, A.R., Madoffe, S., Munishi, P.K.T., Morse-Jones, S., Mwakalila, S., Paavola, J., Naidoo, R., Ricketts, R., Rouget, M., Willcock, S., White, S., Balmford, A., 2011. Measuring, modeling and mapping ecosystem services in the Eastern Arc Mountains of Tanzania. *Progr. Phys. Geogr.* 35, 595–611.
- Freeman III, A.M., 2003. *The Measurement of Environmental and Resource Values: Theory and Methods*, Resources For the Future Press, Washington, DC.
- Glenk, K., Colombo, S., 2011. How sure can you be? A framework for considering delivery uncertainty in benefit assessments based on stated preference methods. *J. Agr. Econ.* 62(1), 25–46.
- Graves, A., Morris, J., 2013. *Restoration of Fenland Peatland under Climate Change*. Cranfield University, Bedford, UK. Report to Adaptation Sub-Committee, London.
- Gollier, C., Koundouri, P., Pantelidis, T., 2008. Declining discount rates: Economic justifications and implications for long-run policy. *Econ. Pol.* 56, 757–795.
- Hanley, N., Schöpfer, F., Spurgeon, J., 2003. Aggregating the benefits of environmental improvements: distance-decay functions for use and non-use values. *Journal of Environmental Management* 68(3), 297–304.
- Hanley, N., Colombo, S., Mason, P., Johns, H., 2007. The reform of support mechanisms for upland farming: paying for public goods in the Severely Disadvantaged Areas of England. *J. Agr. Econ.* 58(3), 433–453.
- Hanley, N., Acs, S., Dallimer, M., Graves, A., Morris, J., Gaston, K., Armsworth, P., 2012. Farm-scale ecological and economic impacts of agricultural change in the uplands. *Land Use Pol.* 29(3), 587–597.

- Hanley, N., Barbier, E.B., 2009. Pricing Nature: cost-benefit analysis and environmental policy, Edward Elgar, Cheltenham.
- Harlow, J., Clarke, S., Phillips, M., Scott, A., 2012. Valuing land-use and management changes in the Keighley & Watersheddies catchment. Natural England Research Report No.44, Peterborough.
- Hodge, I., Reader, M., 2010. The introduction of entry level stewardship in England: extension or dilution in agri-environment policy? *Land Use Pol.* 27(2), 270–282.
- Holden, J., Chapman, P.J., Labadz, J.C., 2004. Artificial drainage of peatlands: hydrological and hydrochemical process and wetland restoration. *Progr. Phys. Geogr.* 28, 95–123.
- Holden, J., 2005. Peatland hydrology and carbon release: why small-scale process matters. *Phil. Trans. Math. Phys. Eng. Sci.* 363, 2891–2913.
- JNCC [Joint Nature Conservation Committee], 2011. Towards an assessment of the state of UK. Peatlands, JNCC report No. 445.
- Jobstvogt, N., Hanley, N., Hynes, S., Kenter, J., Witte, U., 2014. Twenty thousand sterling under the sea: Estimating the value of protecting deep-sea biodiversity. *Ecol. Econ.* 97, 10–19.
- Jørgensen, S.L., Olsen, S.B., Ladenburg, J., Martinsen, L., Svenningsen, S.R., Hasler, B., 2012. Spatially induced disparities in users' and non-users' WTP for water quality improvements : Testing the effect of multiple substitutes and distance decay. *Ecol. Econ.* 92(8), 55–66.
- Joosten, H., Tapio-Bistrom, M.-L., Tol, S., (Eds.) 2012. Mitigation Of Climate Change in Agriculture Series 5: Peatlands - Guidance For Climate Change Mitigation By Conservation, Rehabilitation And Sustainable Use. FAO and Wetlands International, Rome.
- Keenleyside, C., Moxey, A., 2011. Review of public funding of peatland management and restoration in the UK. IUCN Peatland Programme. <http://www.iucn-uk-peatlandprogramme.org/sites/all/files/Review%20Public%20Funding%20of%20Peatland%20Management%20and%20Restoration,%20June%202011%20Final.pdf> (accessed 13 February 2014).
- Knight, F.H., 1921. Risk, Uncertainty and Profit, Houghton Mifflin Company, Boston.
- Krutilla, K., 2005. Using the Kaldor-Hicks Tableau format for cost-benefit analysis and policy evaluation. *J. Pol. Anal. Manag.* 24(4), 864–875.
- Labadz, J., Allott, T., Evans, M., Butcher, D., Billett, M., Stainer, S., Yallop, A., Jones, P., Innerdale, M., Harmon, N., Maher, K., Bradbury, R., Mount, D., O'Brien, H., Hart, R., 2010. Peatland hydrology. Draft report to IUCN UK Peatland Programme, Edinburgh. [http://www.iucn-uk-peatlandprogramme.org/sites/all/files/Review%20Peatland%20Hydrology,%20June%202011%20Draft\\_0.pdf](http://www.iucn-uk-peatlandprogramme.org/sites/all/files/Review%20Peatland%20Hydrology,%20June%202011%20Draft_0.pdf) (accessed 13 February 2014).



- Littlewood, N., Anderson, P., Artz, R., Bragg, O., Lunt, P. & Marrs, R., 2010. Peatland biodiversity. Report to IUCN UK Peatland Programme, Edinburgh. <http://www.iucn-uk-peatlandprogramme.org/sites/all/files/Review%20Peatland%20Biodiversity,%20June%202011%20Final.pdf> (accessed 13 February 2014).
- Lunt, P., Allot, T., Anderson, P., Buckler, M., Coupar, A., Jones, P., Labadz, J. & Worrall, P., Evans, M., 2010. Peatland Restoration. IUCN Peatland Programme. <http://www.iucn-uk-peatlandprogramme.org/sites/all/files/Review%20Peatland%20Restoration,%20June%202011%20Final.pdf> (accessed 13 February 2014).
- Mace, G.M., Norris, K., Fitter, A.H., 2011. Biodiversity and ecosystem services: A multilayered relationship. *Trends Ecol. Evol.* 27, 19–26.
- Martin-Ortega, J., Allott, T.E.H., Glenk, K., and Schaafsma, M., this issue. Integrating hydrological and economic knowledge to value water quality improvements from peatland restoration: evidence and challenges. *Ecosystem Services* (submitted).
- Moran, D., Wreford, A., Evans, A., Fox, N., Glenk, K., Hutchings, M., McCracken, D., McVittie, A., Mitchell, M., Moxey, A., Topp, K., Wall, E., 2013. Assessing the preparedness of England's natural resources for a changing climate: Assessing the type and level of adaptation action required to address climate risks in the 'vulnerability hotspots', Report to the Adaptation Sub Committee, Committee on Climate Change. [http://www.theccc.org.uk/wp-content/uploads/2013/07/Final-report-SRUC-ASC\\_-4-July\\_ASC-FINAL-9-July-2013\\_clean.pdf](http://www.theccc.org.uk/wp-content/uploads/2013/07/Final-report-SRUC-ASC_-4-July_ASC-FINAL-9-July-2013_clean.pdf) (accessed 13 February 2014).
- Moreno-Mateos, D., Power, M.E., Comín, F.A., Yockteng, R., 2012. Structural and Functional Loss in Restored Wetland Ecosystems. *PLoS Biol.* 10(1), e1001247. doi:10.1371/journal.pbio.1001247.
- Morris J., Graves, A., Angus, A., Hess, T., Lawson, C., Camino, M., Truckell, I., Holman, I., 2010. Restoration of Lowland Peatland in England and Impacts on Food Production and Security. Unpublished Report to Natural England. Cranfield University, Bedford.
- Moxey, A., 2011. Illustrative Economics of Peatland Restoration. Technical report to the IUCN Peatland Programme. <http://www.iucn-uk-peatlandprogramme.org/sites/all/files/Illustrative%20Economics%20of%20Peatland%20Restoration,%20June%202011%20Final.pdf> (accessed 13 February 2014).
- Natural England, 2010a. Higher Level Stewardship: Environmental Stewardship Handbook, Third Edition, Natural England, Peterborough.
- Natural England, 2010b. A cost-benefit analysis of peatland restoration options. Unpublished technical report, Natural England.
- Nordhaus, W., 2008. *A Question of Balance: Weighing the Options on Global Warming Policy*, Yale University Press, New Haven.

- Parry, L.E., Holden, J., Chapman, P.J., 2014. Restoration of blanket peatlands. *J. Environ. Manage.* 133, 193–205.
- Pattison, J., Boxall, P.C., Adamowicz, W.L., 2011. The economic benefits of wetland retention and restoration in Manitoba. *Can. J. Agr. Econ.* 59, 223–244.
- Peacock, M., Evans, C.D., Fenner, N., Freeman, C., 2013. Natural revegetation of bog pools after peatland restoration involving ditch blocking—The influence of pool depth and implications for carbon cycling. *Ecol. Eng.* 57, 297–301.
- Pearce, D.W., Atkinson, G., Mourato, S., 2006. Cost-benefit analysis and the environment: recent developments. Organisation for Co-operation and Development (OECD), Paris.
- Pettinotti, L., 2013. Restoration of degraded peatlands in Scotland can generally be justified in economic terms on the basis of GHG emission savings alone. <http://www.teebweb.org/economic-merits-of-peatlands-restoration-in-scotland/> (accessed 13 February 2014).
- Quick, T., Reed, M., Smyth, M., Birnie, R., Bain, C., Rowcroft, P., White, A., 2013. Developing place-based approaches for Payments for Ecosystem Services, URS London.
- Quin, S.L.O., Artz, R.R.E., Coupar, A.M., Littlewood, N.A., Woodin, S.J., 2014. Restoration of upland heath from a graminoid- to a *Calluna vulgaris*-dominated community provides a carbon benefit. *Agri. Ecosyst. Environ.* 185, 133–143.
- Rawlins, A., Morris, J., 2010. Social and economic aspects of peatland management in Northern Europe, with particular reference to the English case. *Geoderma* 154, 242–251.
- Reed, M.S., Moxey, A., Prager, K., Hanley, N., Skates, J., Evans, C., Glenk, K., Thompson, K., this issue. Payment by potential results: paying for ecosystem services in agri-environment schemes in UK peatlands. *Ecosystem Services* (submitted).
- Reed, M.S., Bonn, A., Evans, C., Joosten, H., Bain, B., Farmer, J., Emmer, I., Couwenberg, J., Moxey, A., Artz, R., Tanneberger, F., von Unger, M., Smyth, M., Birnie, R., Inman, I., Smith, S., Quick, T., Cowap, C., Prior, S., Lindsay, R.A., 2013. Peatland Code Research Project Final Report. Defra, London.
- Rigby, D., Alcon, F., Burton, M., 2010. Supply uncertainty and the economic value of irrigation water. *Eur. Rev. Agr. Econ.* 37, 97–117.
- Roberts, D.C., Boyer, T.A., Lusk, J.L., 2008. Preferences for environmental quality under uncertainty. *Ecol. Econ.* 66, 584–593.
- Samples, K.C., Dixon, J.A., Gowen, M.M., 1986. Information disclosure and endangered species valuation. *Land Econ.* 62, 306–312.

- Schaafsma, M., 2010. Spatial effects in Stated Preference Studies for Environmental Valuation, PhD VU University Amsterdam.
- Schaafsma, M., Brouwer, R., Rose, J., 2012. Directional heterogeneity in WTP models. *Ecol. Econ.* 79, 21–31.
- Schaafsma, M., Brouwer, R., Gilbert, A., van den Bergh, J., Wagtendonk, A., 2013. Estimation of distance-decay functions to account for substitution and spatial heterogeneity in stated preference research. *Land Econ.* 89(3), 514–537.
- Strange, N., Jacobson, J.B., Thorsen, B.J., 2007. Value for money: protecting endangered species on Danish heathland. *Environ. Manag.* 40, 761–774.
- Tanneberger, F., Wichtmann, W., (Eds.) 2011. Carbon credits from peatland rewetting. Climate, biodiversity, land use, Schweizerbart Science Publishers, Stuttgart.
- TEEB, 2010. The Economics of Ecosystems and Biodiversity: Ecological and Economic Foundations, Earthscan, London and Washington.
- Tinch, D., Hanley, N., Beharry-Borg, N., 2010. Mountains, Moorlands and Heaths, 2<sup>nd</sup> Draft Economics group report. UK National Ecosystems Assessment.
- Tol, R.S.J., 2005. The marginal damage costs of carbon dioxide emissions: an assessment of the uncertainties. *Energ. Pol.* 33(16), 2064–2074.
- Tol, R.S.J., 2010. The Economic Impact of Climate Change. *Perspektiven der Wirtschaftspolitik* 11, 13–37.
- Turner, R.K., Morse-Jones, S., Fisher, B., 2010. Ecosystem valuation: A sequential decision support system and quality assessment issues. *Ann. New York Acad. Sci.* 1185, 79–101.
- van der Wal, R., Bonn, A., Monteith, D., Reed, M., et al. 2011. Mountains, moorlands and heaths [chapter 5], in: UK National Ecosystem Assessment. Understanding nature's value to society. Technical Report. UNEP-WCMC, Cambridge, 105–160.
- Whitby, M., 2000. Challenges and options for the UK agri-environment: Presidential address. *J. Agr. Econ.* 51(3), 317–332.
- White, P.C.L., Lovett, J.C., 1999. Public preferences and willingness to pay for nature conservation in the North York Moors National Park, U.K. *J. Environ. Manag.* 55, 1–13.
- Wichmann, S., Brander, L., Schäfer, A., Schaafsma, M., van Beukering, P., Tinch, D., Bonn, A., in press. Valuing peatland ecosystem services, in: Bonn, A., Allott, T., Evans, M., Joosten, H., Stoneman, R. (Eds.), *Peatland restoration for ecosystem services*. Cambridge University Press, Cambridge, UK.

Wilson, L., Wilson, J., Holden, J., Johnstone, I., Armstrong, A., Morris, M., 2011. Ditch blocking, water chemistry and organic carbon flux: Evidence that blanket bog restoration reduces erosion and fluvial carbon loss. *Sci. Total Environ.* 409(11), 2010–2018.

Worrall, F., Bell, M.J., Bhogal, A., 2010. Assessing the probability of carbon and greenhouse gas benefit from the management of peat soils. *Sci. Total Environ.* 408, 2657–2666.

Worrall, F., Chapman, P., Holden, J., Evans, C., Artz, R., Smith, P., Grayson, R., 2011. A review of current evidence on carbon fluxes and greenhouse gas emissions from UK peatland. JNCC Report, No. 442.

Yang, W., Wang, X., Liu, Y., Gabor, S., Boychuk, L., Badiou, P., 2010. Simulated environmental effects of wetland restoration scenarios in a typical Canadian prairie watershed. *Wetlands Ecol. Manag.* 18(3), 269–279.